PB86-111572

Strategy for Monitoring of Contaminant Distributions Resulting from Proposed Sewage Sludge Disposal at the 106-Mile Ocean Disposal Site

(U.S.) Environmental Research Lab.
Narragansett, RI

1985

U.S. Department of Commerce
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TECHNICAL REPORT DATA
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1. REPORT NO.	2.	3. R PB86-111572
EPA/600/J-85/173		

4. TITLE AND SUBTITLE	5. REPORT DATE
A STRATEGY FOR MONITORING OF CONTAMINANT DISTRIBUTIONS RESULTING FROM PROPOSED SEWAGE SLUDGE DISPOSAL AT THE 106-MILE OCEAN DISPOSAL SITE (Journal Version)	1985
	6. PERFORMING ORGANIZATION CODE
7. AUTHOR(S) T.P. O'Connor[1], H.A. Walker[2], J.F. Paul[3], and V.J. Bierman,Jr[3]	8. PERFORMING ORGANIZATION REPORT NO. ERLN-652
9. PERFORMING ORGANIZATION NAME AND ADDRESS (1)N/OMS32, NOAA, Rockville, MD,(2)Computer Sci. Corp. C/O ERL-Narra.,(3)U.S.EPA, ERL-Narragansett, South Ferry Rd., Narragansett, RI 02882	10. PROGRAM ELEMENT NO. AARB1A
	11. CONTRACT/GRANT NO. NA
12. SPONSORING AGENCY NAME AND ADDRESS U.S.EPA, ERL-Narragansett South Ferry Road Narragansett, RI 02882	13. TYPE OF REPORT AND PERIOD COVERED Journal Article
	14. SPONSORING AGENCY CODE EPA/600/05

15. SUPPLEMENTARY NOTES

Marine Environmental Research, Vol.16,pp.127-150,1985

16. ABSTRACT

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17. KEY WORDS AND DOCUMENT ANALYSIS

a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group

18. DISTRIBUTION STATEMENT	19. SECURITY CLASS *(This Report)*	21. NO. OF PAGES
DISTRIBUTE TO PUBLIC	UNCLASSIFIED	27
	20. SECURITY CLASS *(This page)*	22. PRICE
	UNCLASSIFIED	9.95

EPA Form 2220-1 (Rev. 4-77) PREVIOUS EDITION IS OBSOLETE

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EPA/600/J-85/173
JOURNAL ARTICLE

Marine Environmental Research **16** (1985) 127–150

A Strategy for Monitoring of Contaminant Distributions Resulting from Proposed Sewage Sludge Disposal at the 106-Mile Ocean Disposal Site

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(Received: 11 April, 1985)

ABSTRACT

It has been proposed that future ocean disposal of sewage sludge from the US east coast be done at a site beyond the edge of the continental shelf. In anticipation of that, a monitoring strategy has been developed to determine the average spatial distribution of contamination. The strategy is an iterative series of measurements developed from models of sludge dispersion and settling which are based on characteristics of the disposal site and sewage sludge. Once disposal is initiated at the site, the strategy requires sampling the upper mixed layer at 36 stations, mostly within 100 km of the site, and deployment of near-bottom sediment traps along a line extending 300 km away from the site. Based on initial results, subsequent sampling locations will be selected to refine estimates of the

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INTRODUCTION

The US Environmental Protection Agency (EPA) designated the 106-Mile Ocean Disposal Site for sewage sludge disposal on a 5-year interim basis (*Federal Register*, 1984*a*). The site is approximately 106 nautical miles* from Ambrose Lightship, over the outer continental slope and continental rise, about 30 nautical miles (54 km) southwest of the mouth of Hudson Canyon, in depths ranging from about 1500 to 2700 m (Ingham *et al.*, 1977) (Fig. 1). Presently sewage sludges from the New York and New Jersey metropolitan area are dumped at a site which is about 12 nautical miles seaward of New York harbor. The 1982 input to the 12-Mile Site was 7·6 million t (R. Reindl, US EPA Region II, pers. comm.). The EPA has initiated the process of removing that site's designation status (*Federal Register*, 1984*b*). Therefore, the possibility that future disposal of sludge will occur at the 106-Mile Site, focuses attention on the need to develop monitoring strategies for deep ocean waste disposal operations.

The 106-Mile Site was designated because of its dispersive characteristics, its remoteness from recreational areas, and its location seaward of the abundant and valued living resources of the continental shelf. The goals of a monitoring plan are to determine the spatial distributions of sludge constituents disposed at the site, and to determine possible biological impacts. Because of the large scales involved at the 106-Mile Site, a phased approach is appropriate, in which information on constituent distributions is acquired first, and then used as a basis for design of the biological monitoring. This paper concerns the first phase of such an overall plan.

The 106-Mile Site lies nominally in the slope water mass but this site can be occupied by shelf water or, during passage of warm core Gulf Stream eddies, Gulf Stream or Sargasso Sea water (Ingham *et al.*, 1977)

* 1 international nautical mile = 1·852 km.

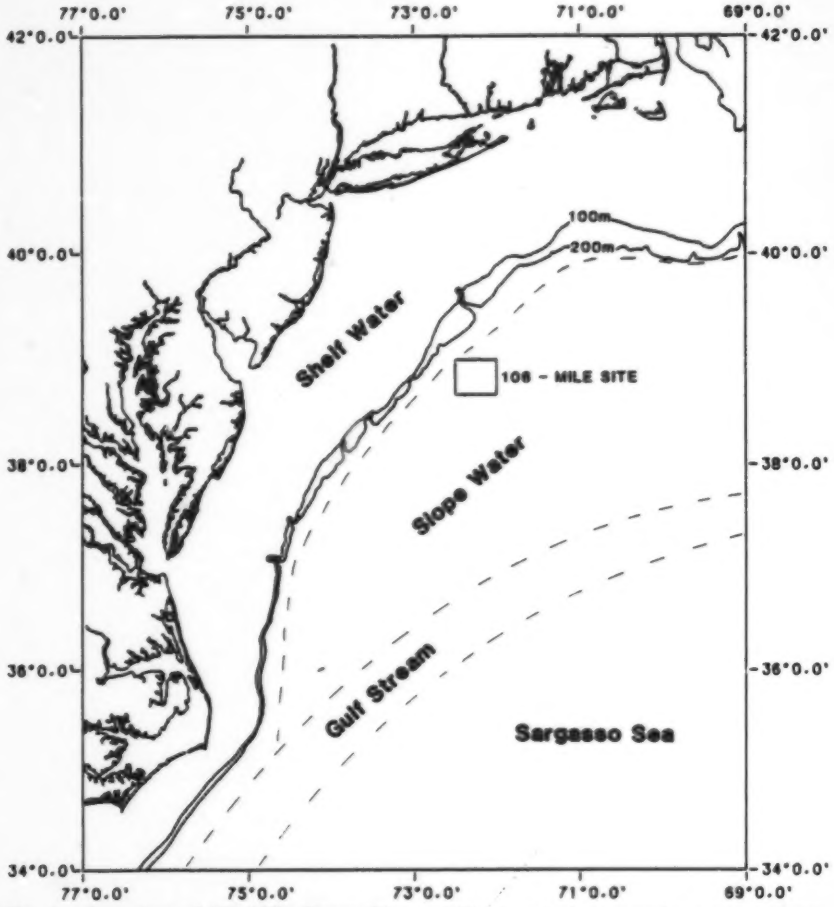

Fig. 1. Location of 106-Mile Ocean Disposal Site in relation to major water masses.

(Fig. 1). The circulation in the region may be quite active. Quantifying physical transport of a waste requires spatial and temporal averaging to characterize some form of mean advective flow and diffusion. In discussing processes affecting the distribution of sewage sludge, we will refer to: (1) small scale processes, those occurring over time scales of 24 h or less and spatial scales of 0 to 10 km; (2) intermediate scale processes, occurring over 1 to 100 days and spatial scales from 10 to 300 km; and (3) large scale processes, important on time scales greater than 100 days and spatially encompassing an ocean basin.

The monitoring plan has been based on information on physical and chemical characteristics of the 106-Mile Site. Existing models have been used to obtain estimates of the distribution of individual sewage sludge constituents in order to strategically locate the sampling sites. The strategy is not intended to sample the rapid, initial near-field dilution that occurs within the first day following discharge of each individual barge load of sludge. Rather, it is designed to quantify distributions of sewage sludge components on intermediate spatial scales. Contamination on this scale will be that due to the integrated effects of daily dumps, not that due to the unique history of any one individual disposal event.

PROCESSES AFFECTING CONTAMINANT DISTRIBUTIONS AT THE 106-MILE SITE

Inertial and tidal motions are advective on small spatial scales, but are considered here to be diffusive mixing processes, because our emphasis is on the intermediate scale. However, effective dispersion at the intermediate scale in slope water depends primarily on motions with time scales greater than 24 h. Warm core Gulf Stream eddies (or rings) fall into this category, since they typically have diameters of 100 km and influence circulation at the disposal site for a period of three weeks (Bisagni, 1976). Thus, warm core rings should receive explicit consideration in a monitoring plan designed for the intermediate scale. Large scale considerations, such as the possibility that the 106-Mile Ocean Disposal site may be embedded in a cyclonic gyre in the slope water (Csanady *et al.*, 1979), may affect contaminant distributions on time scales greater than 1 year. It is recognized that this potential gyre does not represent a closed system, and that the Gulf Stream probably represents the principal downstream sink for much of the relatively fresh water that enters the Northern Slope Water region from the Gulf of Maine (Beardsley & Boicourt, 1981). However, until contaminant distributions can be measured on the intermediate scale, larger scale monitoring cannot be credibly designed.

MODELS PREDICTING PHYSICAL TRANSPORT OF SEWAGE SLUDGE CONSTITUENTS AT THE 106-MILE SITE

A number of investigators have analyzed data from dye studies or waste releases in conjunction with short term dispersion models (Okubo, 1971; Kohn & Rowe, 1976; Orr & Hess, 1978; Bisagni, 1977; Csanady, 1981;

Simpson *et al.*, 1981). While the results of these investigations give first order estimates of plume dilution rates, they are not appropriate on time scales greater than one week.

Two simple, physically reasonable models have been proposed for predicting probable concentrations of sludge constituents on intermediate spatial scales resulting from proposed sewage sludge disposal at the 106-Mile Site (O'Connor *et al.*, 1983; Paul *et al.*, in press). Rather than predicting transport and dispersion of individual dumps of material, projections were made of the average distribution of sludge constituents. Since sewage sludge is a near neutral density waste which would be released slowly from moving barges, a major portion of the material may stay in the upper mixed layer, which, at the 106-Mile Site, is about 20 m during the summer and 100 m during the winter.

The O'Connor *et al.* (1983) model considered the distribution of sludge constituents in the upper mixed layer, and the possible benthic contamination that could result from sludge disposal. Results, based on the composition of New York/New Jersey sewage sludges (Table 1), a sludge input rate of $2 \times 10^4\,m^3\,day^{-1}$ and an advective flux of water through the disposal site in the upper 20 m of $1 \times 10^{10}\,m^3\,day^{-1}$, are listed in Table 2. A sludge dilution factor of 5×10^5 is predicted for the upper mixed layer, which is the ratio of the average advective flux of water in the upper 20 m, to the sludge input rate. Benthic contamination was estimated (Table 3) by allowing for gravitational descent of some sewage particles, horizontal dispersion of particles below the upper mixed layer, and the influence of bioturbation on the seafloor. There are a number of assumptions behind these estimated concentration distributions. For example, water-borne sludge is diluted only with water that flows directly through the disposal site. The influence of warm-core Gulf Stream eddies on enhancing dispersion in the upper mixed layer has not been explicitly included. Twenty percent of the sludge solids is estimated to fall at $10\,m\,day^{-1}$ ($0{\cdot}01\,cm\,s^{-1}$) or faster, with the remainder having slower settling velocities. Horizontal dispersion is assumed to proceed below the upper mixed layer at only 10% of its rate in the upper ocean. For the most part, all of these assumptions tend to cause predictions for contaminant concentrations at the intermediate scale to be higher than actually may occur. There is still considerable uncertainty about sludge settling. It is possible that more than 20% of the sludge may descend more rapidly than at $10\,m\,day^{-1}$. If this is the case, the O'Connor *et al.* (1983) model may underestimate benthic contamination.

Paul *et al.* (in press) have made a more quantitative estimate of water

TABLE 1
New York/New Jersey Sewage Sludge Characteristics

Component	*Concentration* ($\mu g\,g^{-1}$)	
	wet	*dry*
Total solids	33 000[a]	—
Suspended solids	39 000[b]	—
Biological oxygen demand	17 000[b]	—
Chemical oxygen demand	93 000[b]	—
Total organic carbon	9 200[b]	180 000
Total nitrogen	1 500[b]	—
Total phosphorus	400[b]	—
Cadmium	1·5	44[a]
Mercury	0·3	8[a]
Copper	54	1 637[a]
Lead	33	985[a]
Zinc	93	2 809[a]
Iron	1 100	22 000[c]
Chromium	26	784[a]
Nickel	7·0	211[a]
Vanadium	0·6	19[a]
Arsenic	0·5	14[a]
Selenium	0·1[b]	2
Total PCBs	0·05	1·5[d]
Total PAHs	0·7	14

[a] EAI & SEAMOcean (1983). Their data for those constituents are weighted averages from 1981 analyses of sludge submitted to EPA by individual plants. Concentrations for PCBs and vanadium in sludges not from New York City were assumed equal to the weighted average for New York City plants. Reported dry weight concentrations converted to wet weight concentrations using a 3·3% total solids concentration.

[b] Data from Mueller *et al.* (1976), who calculated weighted averages from 1973 analytical data. The selenium concentration is 1000 times less than that reported (incorrectly) by Mueller *et al.* (1976). The obvious discrepancy of the total solid concentrations being less than suspended solids is of no consequence in this report. Reported wet weight concentrations converted to dry weight concentrations using a 5% total solids concentration. Where both EAI & SEAMOcean (1983) and Mueller *et al.* (1976) report concentrations the more recent concentrations are lower. The factors by which they are lower on a dry weight basis are as follows: TS (1·35), Cd (1·68), Cu (0·73, i.e. higher now), Pb (1·28), Zn (1·14), Cr (1·61), Ni (0·90), V (0·84), As (0·14). O'Connor *et al.* (1983) used a Hg concentration which was half of the Mueller *et al.* (1976) value and 1·25 times higher than the EAI & SEAMOcean (1983) value.

[c] Data for iron is from Furr *et al.* (1976).

[d] MacLeod *et al.* (1981) analyzed New York City sludge composites for organic compounds. The PAH concentration given here applies to 3-ring or larger compounds. Sludge also contained 0·3 $ng\,g^{-1}$ naphthalenes and 0·2 $ng\,g^{-1}$ biphenyls. There is a large discrepancy between the 1·5 $\mu g\,g^{-1}$ (dry) PCB concentration from EAI and SEAMOcean (1983) and the 0·4 $\mu g\,g^{-1}$ (wet) ($= 12{\cdot}1\ \mu g\,g^{-1}$ (dry) at 3·3% total solids) reported by MacLeod *et al.* (1981). The lower value is used in this discussion, but this point deserves further attention. The lower value is closer to the value of 3·5 $ng\,g^{-1}$ (dry) reported by West & Hatcher (1980) and by Bopp *et al.* (1981).

TABLE 2
Comparisons of Sludge Constituent Concentrations with Ambient Surface Water Concentrations

Sludge component	*(A) Concentration ($ng\,kg^{-1}$)[a] after dilution*	*(B) Ambient[b] concentration ($ng\,kg^{-1}$)*	$\frac{(A)+(B)}{(B)}$
Suspended solids	$7{\cdot}8 \times 10^4$	$(10 \text{ to } 30) \times 10^4$	1·8–1·3
Oxygen demand (O_2 loss)	$1{\cdot}9 \times 10^5$	$(43 \text{ to } 70) \times 10^5$ (O_2)	1·04–1·03
Total organic carbon	$1{\cdot}8 \times 10^4$	100×10^4	1·02
Total nitrogen	1 200	9 700 (summer)	1·1
Total phosphorus	800	31 000 (summer)	1·02
Iron	2 200	500	5·4
Cadmium	3·0	14	1·2
Mercury	0·6	5	1·1
Copper	108	170	1·6
Lead	66	30	3·2
Zinc	186	30	7·2
Chromium	52	70	1·6
Nickel	14	200	1·1
Vanadium	1·2	1 500	1·001
Arsenic	1·0	90	1·01
Selenium	0·2	40	1·005
Total PCBs	0·1	0·05	3·0
Total PAHs	0·7	—	—

[a] Concentrations are those in Table 2 after dilution by a factor of 5×10^5, which is the ratio of the average advective flux of water ($1 \times 10^{10}\,m^3\,day^{-1}$) in the upper 20 m to the sludge input rate of $2 \times 10^4\,m^3\,day^{-1}$.

[b] Data for ambient surface water concentrations of suspended solids, iron, cadmium, and copper are from Kester *et al.* (1981). Dissolved oxygen data are from Warsh (1975). Nitrogen and phosphorus concentrations are averages from Peterson (1975). All the above concentrations were measured in slope water in the context of 106-Mile Site investigations. The remaining concentrations are for open ocean surface waters; mercury (Mukherji & Kester, 1980), nickel (Bruland *et al.*, 1979), chromium (Cranston & Murray, 1978), arsenic (Andreae, 1979), selenium (Measures *et al.*, 1980), and vanadium (Brewer, 1975). Ambient concentrations of three possible tracers of sludge have been revised since this table first appeared in O'Connor *et al.* (1983). The zinc concentration listed here is that reported by Bruland & Franks (1983) for slope water. The lead value is that for slope water reported by Boyle & Huested (1983) and is similar to that of Schaule & Patterson (1983) for Sargasso Sea water. The PCB concentration is that for the outer New York Bight reported by Boehm (1983).

TABLE 3

Sediment Contamination Due to a Sludge Flux Equivalent to 20% of One Year's Total Solids Settling over 10^4 km^2 and Mixing into Sediment with a Bioturbation Rate of 0·1 cm^2 year^{-1}. All Concentrations in Units of μg g^{-1} (Dry)[a]

Sludge component	*Maximum*[b] *conc. increase after 1 year*	*Concentration increase observed*[c] *if upper 1 cm of sediment is homogenized for analysis after*			*Ambient*[d] *conc.*
		1 year	*10 years*	*100 years*	
Cadmium	0·07	0·008	0·15	0·60	1·3
Mercury	0·02	0·001	0·09	0·17	0·2
Copper	2·3	0·28	5·0	19	25
Lead	1·5	0·17	3·3	13	15
Zinc	4·3	0·48	9·5	37	49
Chromium	1·2	0·14	2·6	10	24
Nickel	0·35	0·04	0·08	3·0	31
Total PCBs	0·002	0·00 3	0·004	0·02	0·001
Total PAHs	0·03	0·004	0·07	0·3	0·05
Total organic content	0·04 (%)	0·005 (%)	0·09 (%)	0·34 (%)	1·0 (%)

[a] Calculations developed in O'Connor *et al.* (1983), dry sediment density of 2·7 g cm^{-3} is assumed.

[b] This concentration applies at the sediment–water interface. Maximum concentrations after any year, x, are these concentrations times $\sqrt{x}$.

[c] Concentrations are not uniform in the upper 1 cm of sediment, but would effectively be constant if the upper 1 cm is mixed during sample collection or analysis.

[d] Elemental concentrations are from Grieg & Wenzloff (1977) for analyses of sediments below the 106-Mile Site and southwest of it. Except for Cd they are lower than concentrations in deep sea clays reported on a carbonate-free basis (Riley & Chester, 1971). Sediments in the slope water region contain about 40% clay and 40% calcium carbonate (Rowe & Clifford, 1978). PCB value is from Harvey & Steinhauer (1976). PAH value (3-ring and larger) is calculated from Laflamme & Hites (1978). Total organic carbon concentration is from Rowe & Cliffford (1978).

column dispersion in the upper mixed layer. They used current meter records from Woods Hole Oceanographic Institution's Site-D (39°20′ N, 70°00′ W) to compute dispersion due to variations about the average flow. They estimated a lower bound on dispersion in the upper mixed layer by allowing current variations to be only 10% of that in the actual current meter records. The results for this low dispersion case and vertical mixing limited to the upper 20 m are similar to those of O'Connor *et al.* (1983). The main difference is that even for a low dispersion calculation, Paul *et al.* (1983) predict a wider horizontal dispersion than O'Connor *et*

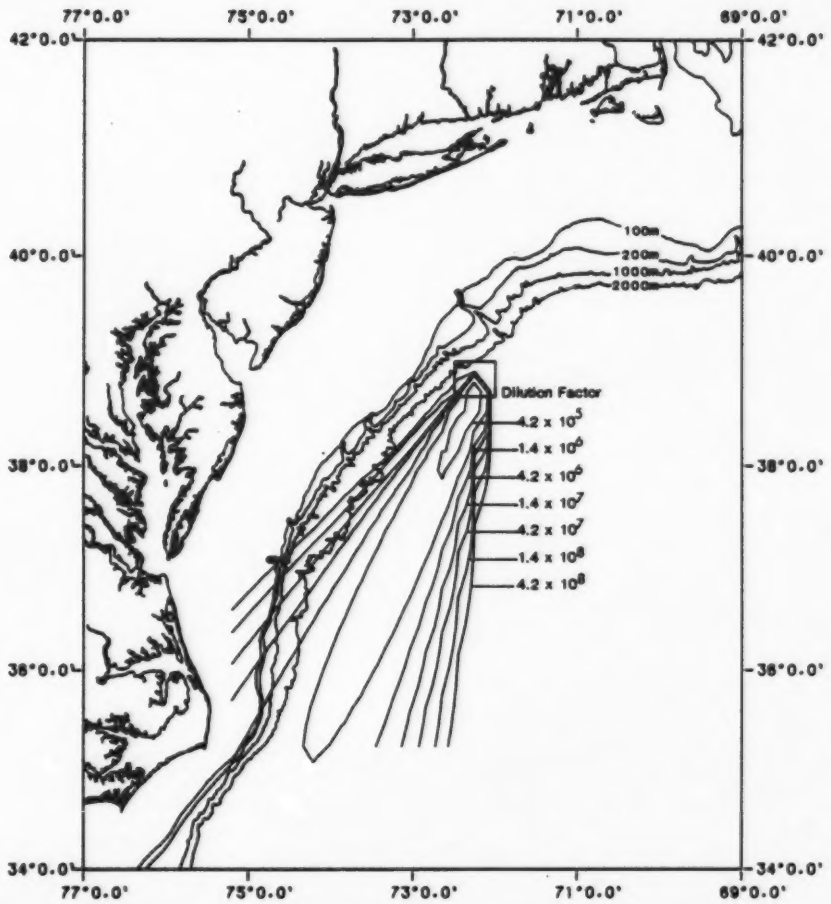

Fig. 2. Distribution of sludge if 7×10^6 m^3 were annually dumped at the 106-Mile Site and subject to 'low' dispersion. Contours are dilution factors and are relabeled from the original contours of Paul *et al.* (in press) which were copper concentrations and assumed a 100 m mixing depth rather than the 20 m depth used here.

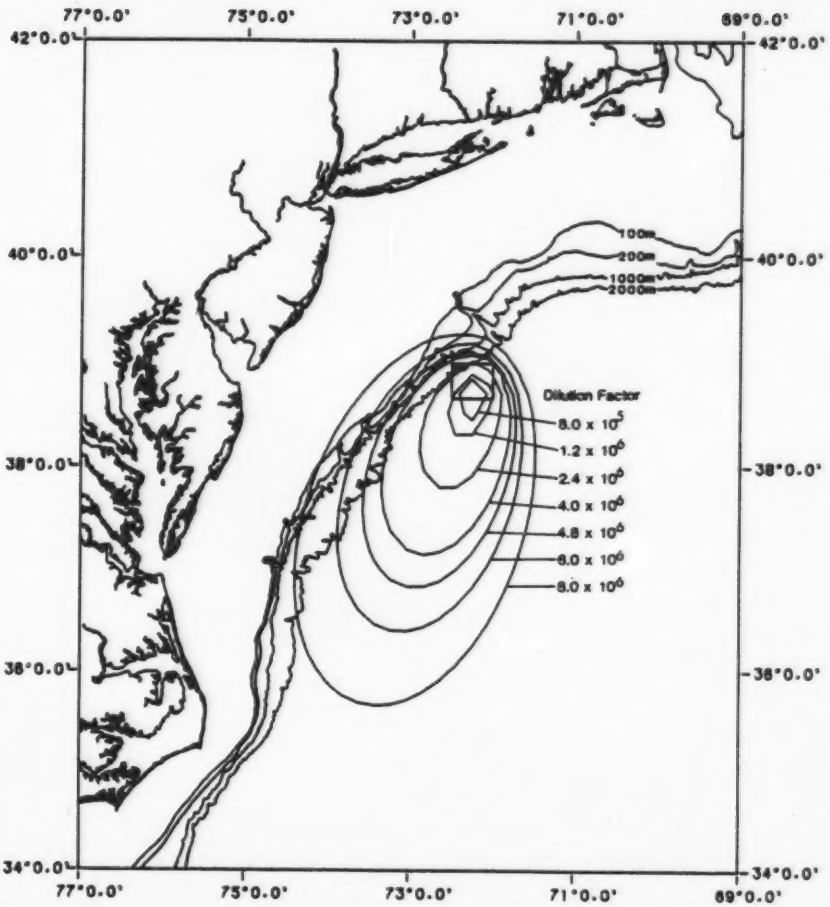

Fig. 3. Similar to Fig. 2 but for 'high' dispersion calculated by allowing all non-periodic variation in current meter record to disperse sludge. Contours are dilution factors and are relabeled from those of Paul *et al.* (1983) which were based on a 100 m, instead of 20 m, mixing depth and which expressed dilution relative to initial diluting volume into which waste is discharged from a vessel.

al. (1983), and dilution by more than the factor of 5×10^5 estimated by O'Connor *et al.* (1983) for the upper mixed layer southwest of the 106-Mile Site (Fig. 2). By allowing for more variation in the current (total variation in record discounted by 40 % corresponding to a low pass filter which removes inertial and tidal periodic motions), their predicted distributions (Fig. 3) become much wider, with dilutions by more than a factor of 10^6 within 10 km of the dumpsite.

As previously stated, the models described by O'Connor *et al.* (1983) and Paul *et al.* (in press) are designed to describe contaminant distribution on intermediate time scales (up to 100 days). Both models are based on the assumption that the Gulf Stream acts as the principal downstream sink for most of the waste material, and thus both implicitly assume that there is no recirculation of sludge constituents back into slope water. Bisagni (1983) has reported the trajectory of five satellite tracked surface drogues which migrated southwestward from the 106-Mile Site to the Gulf Stream. None of them returned to slope water. Conceptually, there could be a slope water gyre which requires a northeast flow parallel to the Gulf Stream with a subsequent southwest flow along the continental slope (Csanady, 1983). In principle this could be incorporated into models of the distribution of sludge. For intermediate scale calculations it is reasonable to consider the Gulf Stream as an ultimate sink.

MONITORING TO DETERMINE CONTAMINANT DISTRIBUTION AT THE 106-MILE SITE

Water column monitoring

The principal objective of monitoring water column contamination is to estimate the distribution of sludge constituents on the intermediate spatial scale. Initial considerations are to find chemical or other indicators of sludge in water as far from the site as possible, and to determine spatial gradients in their concentration consistent with a source at the site.

The modeling indicates that sludge will be extensively diluted with the highest levels of contamination tending away from the dumpsite towards the southwest. Within 20 km of the site, and under low dispersion conditions, sludge constituents in the mixed layer should be diluted

relative to concentrations in the original sludge by a factor of 5×10^5. Monitoring requires selection of sludge indicators which are measurable at that dilution and distinctly elevated above ambient concentrations.

A comparison between diluted sludge and ambient seawater (Table 2) indicates that one cannot expect to find sludge (except close to discharging vessels) by determinations of nutrients, total organic carbon, total suspended solids or most trace elements. Assuming natural variations in concentration to be a factor of 2, sludge additions to total concentrations of these constituents will not be evident above background. Exceptions are zinc, iron, and lead, which at a dilution of 5×10^5, would exceed background concentrations by factors of 7, 5 and 3, respectively. Of the three elemental indicators, zinc is probably the best because it should have the highest expected increase over background, and because there is no other significant source of the element to the disposal site. There is a practical disadvantage to using zinc as a sludge tracer. To avoid contamination due to the sampling ship itself, it has been necessary to sample away from the ship in a raft (Spencer *et al.*, 1982; Bruland & Franks, 1983). There is an additional source of iron input from acid–iron waste now being disposed at the 106-Mile Site. However, on a mass basis, sludge inputs of iron would overwhelm the signal due to acid–iron waste disposal, and thus, iron may be useful as a sludge indicator. Although it is possible to use lead as a sludge tracer, analyses for this metal are certainly not less subject to contamination than zinc and probably are more difficult to perform correctly (Schaule & Patterson, 1983).

There may be elements, not listed in Table 2, which could serve as sludge tracers. However, the most useful tracers could be sludge constituents which, except through human activity, would not be found at all in the ocean. Among such constituents are synthetic organic compounds. Polychlorinated biphenyls (PCBs) should be good indicators when present in sludge at concentrations of tenths of a $mg\,kg^{-1}$. Within the New York Bight, Boehm (1983) found PCB concentrations in seawater of less than $0{\cdot}1\,ng\,kg^{-1}$; in open ocean water levels should be even lower. Thus, until sludge is diluted by more than 10^6 times, PCB concentrations could be indicative of sludge.

Another useful organic compound is the steroid coprostanol, which is produced only by terrestrial mammals and enters the sea only as fecal material (Hatcher & McGillivary, 1979). R. Eganhouse (pers. comm.) has estimated that sludge diluted by a factor of 10^6 should display a

coprostanol concentration of about 10 ng kg^{-1}. There have been no open ocean measurements of coprostanol concentration, but Boehm (1983) found concentrations of 1 to 3 ng kg^{-1} seaward of the sludge disposal site in New York Bight. Further seaward the 'background' should decrease to less than that. Therefore, coprostanol should be a good indicator of sludge up to dilutions of 10^6. Eganhouse & Kaplan (1984) have demonstrated the presence of municipal wastes in coastal sediments via analyses for linear alkyl benzenes and alpha-tocophenyl acetate. These and possibly other man-made compounds might prove to be valuable tracers of sludge in the water column.

A final category of sludge indicator includes microbes which are unique to the feces of humans and other mammals. Spores of the bacterium *Clostridium perfringens* have been used by Cabelli & Pederson (1982) to trace the benthic area influenced by sewage in the New York Bight. Its major advantage over other bacterial or viral indicators of terrestrial fecal matter is its temporal stability. Whereas with other microbes one has to consider 'die-off' in seawater, *C. perfringens* is a conservative tracer. EAI & SEAMOcean (1983) report that New York City sludges contain (1 to 4) $\times 10^7$ *C. perfringens* per 100 ml, so at a dilution of 10^6, the *C. perfringens* concentration would be 10 to 40 per 100 ml. Concentrations of 1 per 100 ml can be routinely detected (V. Cabelli, pers. comm.).

The water column monitoring is based on detecting chemical and microbiological tracers of sludge in water samples collected over a 36 station grid (Fig. 4). This grid emphasizes sampling over the shelf edge and continental slope south of the disposal site. To maximize the chances of detecting sludge constituents, monitoring should be done between May and September. During this period the seasonal pycnocline will inhibit mixing to below about 20 m, whereas during winter the upper mixed layer will be 3 to 7 times deeper. If during a particular cruise, it is known from satellite imagery that the site is within or influenced by a warm-core Gulf Stream eddy, some of the southern far-field stations can be replaced by others north and northwest of the disposal site. This station grid will allow delineation of water contamination up to 100 km from the disposal site. If sludge contamination is detected at that distance the station grid can be expanded. If sludge contamination is not detected at the stations farthest from the 106-Mile Site, possibly due to larger than anticipated vertical losses resulting from particulate settlement, the station grid can be contracted.

The sampling grid for the upper water column reflects an attempt to

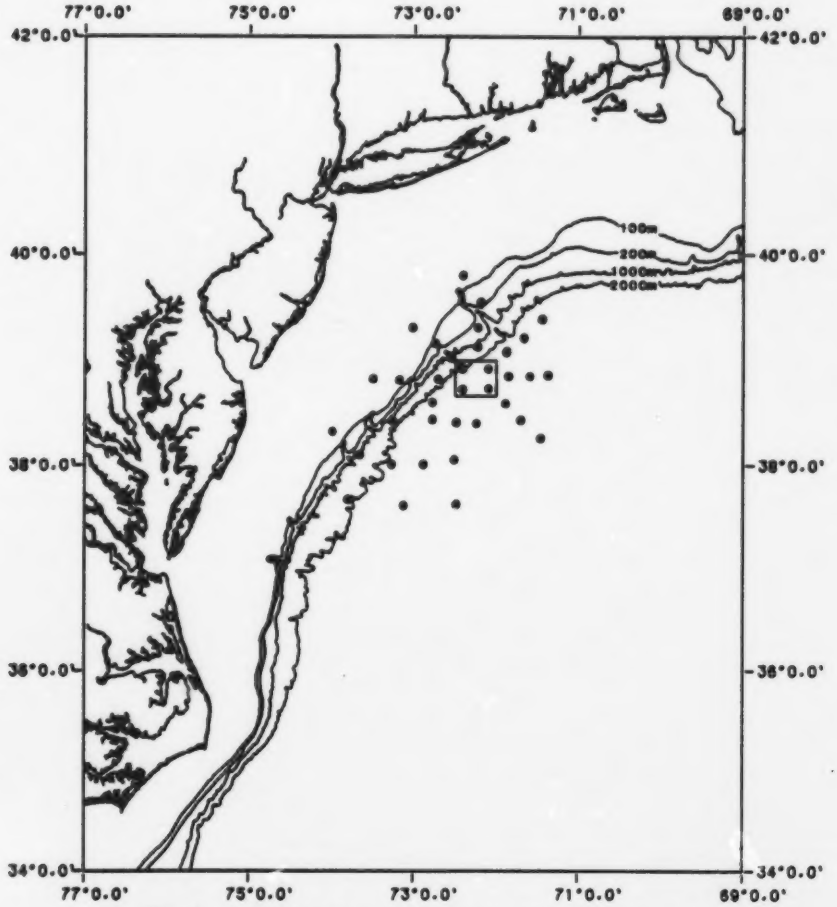

Fig. 4. Initial locations for 36 water sampling stations.

estimate gradients in sludge constituent concentration, with samples spaced more closely in regions of expected steeper gradients. The plan is a modification of a design proposed by Saila *et al.* (1977) for sampling selected trace elements in New York Bight sediments in the vicinity of the 12-Mile Site. This design is more efficient for estimating distributions of selected sludge tracers than a sampling grid with stations spaced at equal increments away from the dumpsite.

At each station, water samples should be taken at depths above and below the seasonal thermocline or permanent pycnocline (in the upper mixed layer, and below it) for each of the principal sludge indicators. A

comparison between samples taken in the mixed layer and below it then can be made to determine if sludge components settle out of the mixed layer. Actual analysis of samples below the mixed layer will only be necessary for those stations where sludge tracers are detected in the upper mixed layer.

To allow interpretation of contaminant data in terms of water mass distributions, routine vertical temperature (XBT) and surface water salinity measurements, selected vertical profiles of salinity and temperature (CTD), and synoptic remote sensing data on sea surface temperature distributions should be obtained. Some measurements of surface and subsurface currents also should be obtained to characterize water motions during sampling periods. Estimated water mass distributions and motions can be used to interpret measured distributions of sludge tracers during different sampling periods.

Strictly speaking, 'control' stations need not be specified since the presence of sludge contamination should be evident if concentration increases towards the disposal site. However, a sound scientific approach would dictate sampling water at a reference site outside the major influence of the dumpsite.

The objective of this monitoring strategy is not to provide a snap-shot of contaminant distribution, but to eventually determine the long term average quasi-steady distribution. During the first year following the initiation of sludge disposal there should be two 15–30 day cruises. Each cruise would sample at each station once every four days (different days for different stations). Since 9 stations must be sampled each day, it may not be possible to deploy and retrieve a raft for zinc sampling at every station. Zinc sampling may be done only at the closely spaced stations. Results from that first year will be two 15–30 day average distributions including a description of the variability in sludge tracer concentrations at each station. Given this data base describing the exposure field, subsequent sampling could seek biological differences between planktonic communities within and outside observed contaminated zones. Measurements of contaminants will also be necessary during these biological surveys; however, the spatial scale of these measurements can probably be reduced, depending on previous results.

Excursions of water masses from the dumpsite to the continental shelf

Since excursions of water masses from the dumpsite to the continental shelf will probably be infrequent and variable in trajectory, consequences

of sludge disposal at the 106-Mile Site to commercial shelf fisheries are expected to be negligible. Given the importance of the shelf, however, it is proposed that a series of satellite tracked drifters be deployed to document the frequency and extent of on-shelf excursions from the 106-Mile Site. From an analysis of available current meter records obtained beyond the 200 m isobath (i.e. the shelf break), Flágg *et al.* (1985) have noted that periods of on-shelf displacements occur with an incidence of roughly 15% at rates which could carry wastes from the disposal site to the outer portions of the continental shelf within a period of several days. However, there is no evidence from the few drifter tracks currently available of any excursions from the 106-Mile Site to shoreward of the shelf break (Bisagni, 1983).

It is recommended that drifters be deployed twice a month (possibly from a ship of opportunity) for two three-month periods (summer and winter) before actual disposal operations commence. This sampling period is longer than estimated Lagrangian decorrelations times (Paul *et al.*, unpublished), and would more than double the number of drifter tracks currently available for estimating Lagrangian transport statistics for the 106-Mile Site. It is also recommended that this effort be continued with deployment of drifters from the sludge disposal barges as part of an ongoing monitoring program for a full year after disposal starts.

Sediment contamination

Water column contamination is non-cumulative. Contamination revealed by water column monitoring will not increase unless the rate of sludge input increases. Sedimentary contamination is predominantly cumulative, and has been related directly to biological effects of sludge disposal (Bascom *et al.*, 1981; Stanford *et al.*, 1981). As discussed previously, O'Connor *et al.* (1983) have calculated rates of accumulation of benthic contamination (Table 3). They argued that sludge deposition would occur over a 10^4 km^2 area downstream of the disposal site.

A rough calculation of where that deposition might occur can be based on an estimate of average water flow as a function of depth and sludge settling characteristics. Ingham *et al.* (1977) have summarized average velocities from long-term current meter records at Site-D. At all depths the average flow is projected to be towards the southwest from the 106-Mile Site in conformance to the bathymetric contours. Average velocities were as follows: 11 cm s^{-1} within the upper 20 m, 6 cm s^{-1} between 20 and

60 m, 8 cm s^{-1} from 60 to 150 m, and 2 cm s^{-1} at all depths below 150 m. If there were no departures from these mean flows, the horizontal distance, D (km), traveled by a sludge particle with a settling velocity, w (cm s^{-1}), can be estimated as: $D = 4{\cdot}88/w$. Thus, for particles with fall velocities of 0·01, 0·1, 1 and 10 cm s^{-1}, the horizontal distance traveled in this mean flow would be 488, 49, 5 and 0·5 km, respectively. If particles fell at 1 cm s^{-1}, one could expect to find them immediately within the disposal site. However, experiments on sludge settling (Faisst, 1981; Hunt & Pandya, 1984) do not find velocities that high. The O'Connor *et al.* (1983) estimate of sludge settling based on data of Faisst (1981), is that 5 % of the solids would settle at 0·1 cm s^{-1} or faster and that 20 % of the solids would fall 0·01 cm s^{-1} or faster. Sludge particles with settling velocities less than 0·01 cm s^{-1} would reach the seafloor only after their incorporation into fast settling biological debris and not contribute significantly to the deposition of sludge within the slope region.

More detailed analysis which accounts for variability in current meter records and sludge flocculation, could produce improved approximations of deposition locations. Oceanographic considerations aside, details of the sludge discharge process are important factors in determining the settling sludge flux. The settling velocity estimates just given are based on the assumption that sludge is initially diluted by a factor of about 1000 by being slowly discharged from a moving vessel. If sludge is discharged quickly there will be less initial dilution and, therefore, more coagulation among sludge particles, yielding an increase in the fraction of total mass incorporated into fast settling flocs.

The monitoring objective is to find where and how much sludge is reaching the seafloor in the slope water region. For two reasons this will require intercepting sludge in sediment traps rather than direct seafloor sampling. First, sludge would be mixed with sediment through bioturbation. Second, regardless of bioturbation, the estimated sludge flux would be only about 10 % of the natural mass flux, which itself results in sediment accumulation rates of only about 0·1 mm $year^{-1}$. In practice, it would not be possible to separate a year's worth of sludge sedimentation from natural sedimentation once it reaches the bottom.

Sediment traps can be placed at any depth. However, the farther from the bottom a trap is placed, the more extrapolation is needed to calculate the area and magnitude of bottom deposition. Since defining bottom deposition is the objective, traps should be set as close as possible to the seafloor. To minimize collection of sediment diluted by resuspended

material, sludge deposition should be monitored by a series of sediment traps set 100 meters above bottom (mab). It is possible that warm-core Gulf Stream eddies could resuspend sediments into the 100 mab zone, and thus bias measurements of deposition rates. However, since the major objective of the initial trap deployments is simply to find out where sludge is reaching the seafloor in the slope water region, some bias in measurement of actual deposition rates may be tolerated.

Trap locations would be determined through an iterative process. Figure 5 is the recommended array for the first deployment of twelve

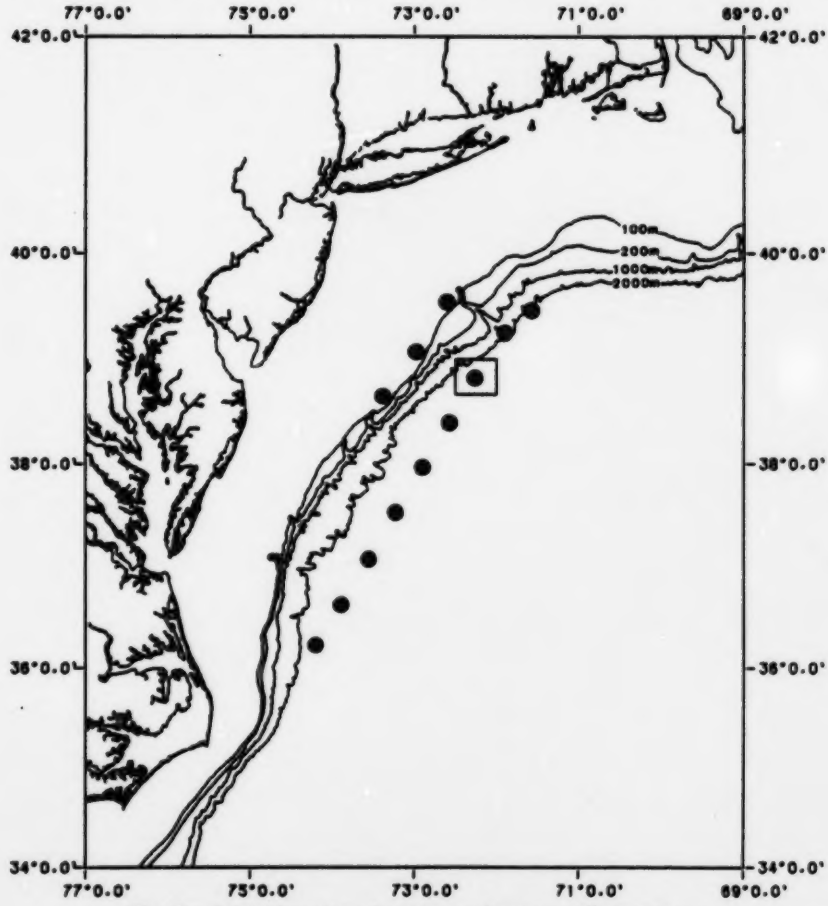

Fig. 5. Initial locations for 12 near-bottom sediment traps.

traps. As for water monitoring, the outer shelf is included in the array because of its important biological resources. Except for three traps on the shelf, the traps form a northeast–southwest line through the disposal site along the contour of the shelf to the area where the Gulf Stream is encountered. This longitudinal array should reveal the distance from the site where deposition is detectable. Armed with data from this array the twelve traps can be redeployed to emphasize detection of the latitudinal distribution of settling sludge.

Detecting sludge in sediment traps requires identifying it in the presence of ambient material. Sedimentation rates within the 106-Mile Site have been measured to be within the range of 350–450 mg m^2 day^{-1} (Rowe & Gardner, 1979; J. Deming, pers. comm.). Comparisons of sludge with natural sediments in the continental slope indicate that both elemental and organic chemical analysis can reveal sludge if it comprises about 10% or more of the material collected in a trap. If sludge is more dilute, only organic chemical or microbial indicators whose ambient concentration is expected to be zero would be useful. Accordingly, the same tracers recommended for water column monitoring (zinc, PCBs, coprostanol, and *C. perfringens*) could be used to identify deposited sludge.

Traps should be moored with acoustical release devices and be retrieved after six months, which should be a long enough period to smooth out variations in sludge settling due to variable currents within a season. Given an objective of detecting a sludge flux of 1 g m^{-2} $year^{-1}$ (about 1% of the ambient mass flux) and a requirement of 1 mg of sludge for analysis of each of four tracers, the minimum trap opening for a six-month deployment would be about 0·01 m^2. Conventional traps with 0·05 m^2 openings would, therefore, be more than adequate.

Once areas receiving sludge are defined, separate studies can be initiated to determine potential biological responses. For example, these studies could seek changes among indigenous benthic communities or establish differences between colonization rates by benthic animals in sediment trays placed within and outside areas receiving sludge.

SUMMARY

A monitoring plan has been designed for determining spatial distributions of sewage sludge constituents in the water column and sediment

resulting from proposed disposal at the 106-Mile Ocean Disposal Site. The plan is based on existing data for physical and chemical characteristics of the 106-Mile Site and of sewage sludges, and on results from existing models for estimated distributions of sludge constituents. The proposed plan is intended to provide information which can be used to design the biological components of an overall monitoring plan.

The emphasis in the water column is on determining sludge indicators as far from the disposal site as possible, and on determining spatial gradients to confirm the source of these indicators. Within 20 km of the dumpsite, under conditions of low dispersion, it is estimated that sludge in the upper mixed layer will be diluted by a factor of 5×10^5, relative to sludge concentrations in the disposal barge. Sludge constituents with the highest probability for detection, relative to ambient levels, are zinc, iron, polychlorinated biphenyls, coprostanol, and spores of the bacterium *C. perfringens*.

The emphasis in the sediment is on determining the locations and quantities of sludge which reach the seafloor in the slope water region. Sediment traps are recommended for determining sludge flux rates and depositional zones. This is because the estimated vertical flux of sludge is a small fraction of the natural sedimentation flux, which itself is very small, and because the sludge constituents will be mixed with natural sediment by bioturbation. The sludge constituents with the highest probability of detection in the sediment traps are the same as the corresponding constituents for the water column.

ACKNOWLEDGEMENTS

Contribution No. 652 of the EPA Environmental Research Laboratory–Narragansett. The authors would like to thank G. Pesch, J. Bonner, J. Heltsche, D. Miller, J. Prager and D. Phelps of the EPA Environmental Research Laboratory–Narragansett and D. Wolfe of the NOAA Ocean Assessment Division for their critical review of the manuscript.

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